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Quantitative review of riparian buffer width guidelines from Canada and the United States

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Abstract

This paper reviewed the provincial, territorial, and state guidelines for the retention of treed riparian buffers after timber harvest in Canada and the United States. Comparisons amongst jurisdictions were facilitated through the use of a standardized template for the classification of waterbodies. Mean buffer widths varied from 15.1 to 29.0 m for different waterbody types when both countries were combined. However, Canadian jurisdictions had wider buffers (except for intermittent streams). In part, this was due to the high percentage of Boreal jurisdictions in Canada and Southeast jurisdictions in the United States. The Boreal region had the widest buffers while Southeastern jurisdictions had the narrowest buffers. Just under half (~44%) of the jurisdictions investigated had three or more modifying factors in the guidelines. Of these, waterbody type, shoreline slope, waterbody size, and presence of fish were the most common. Boreal and Pacific jurisdictions tended to have a more diverse set of waterbody size classes, waterbody types, and other modifying factors. Jurisdictions from the Midwest, Northeast, and Southeast maintained relatively simple 'one-size-fits-all' guidelines. Jurisdictions without modifying factors for slope or presence of fish applied wider baseline buffers than jurisdictions with these factors. A large percentage of jurisdictions (~80%) allowed some selective harvest in buffers. However, these were often accompanied by relatively restrictive prescriptions. In comparison to the ecological recommendations, buffer widths for most jurisdictions were adequate to protect the aquatic biota and habitats but were, generally, less than recommended widths for terrestrial communities. In the future, two management trends are likely to continue, the shift towards more complicated guidelines and the expansion to larger-scale, watershed planning of riparian areas.

Keywords: Riparian; Management; Buffers; Forestry; Streamside; Guidelines; Canada; United States

1. Introduction

The diversity of biota in riparian areas reflects a spatially and temporally heterogeneous environment created by the varied processes affecting the riparian ecosystem. These include fluvial disturbances (flooding, erosion, sedimentation, geomorphic channel processes), non-fluvial disturbances (fire, insects, wind), variable light environment, variable soils, variable topography, and other upland influences (Gregory et al., 1991; Naiman et al., 1993; Sagers and Lyon, 1997). Understanding the spatial extent of these processes is a critical component of riparian management. The riparian zone can be examined along three spatial axes. These include: longitudinal, vertical, and transverse (after Malanson, 1993; United States Fish and Wildlife Service, 1997). Most of the past and present research and management efforts focus on the transverse properties of riparian areas, particularly its translation into buffers left after harvest. The retention of buffers has been recommended for controlling erosion and sedimentation (Haupt and Kid, 1965; Patric, 1978; Moring, 1982; but see Steedman and France, 2000), moderating stream temperature and light (Brown, 1969; Helvey, 1972; Aubertin and Patric, 1974; Beschta and Weatherred, 1984; Kochenderfer et al., 1997; Johnson and Jones, 2000), inputting fine and large organic debris (Murphy and Koski, 1989; McDade et al., 1990; Robinson and Beschta, 1990; Van Sickle and Gregory, 1990; Bilby and Bisson, 1992; Duncan and Brusven, 1985; France et al., 1996;

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Reid and Hilton, 1998; Hauer et al., 1999), and maintaining invertebrate communities (Newbold et al., 1980; Noel et al., 1986; Carlson et al., 1990; Collier and Smith, 1998; Rask et al., 1998; Whitaker et al., 2000), fish communities (Wesche et al., 1987; Young et al., 1999), nearshore vegetation (Johnson and Brown, 1990; Darveau et al., 1998; Harper and MacDonald, 2001), bird communities (Gilmer et al., 1978; Johnson and Brown, 1990; Darveau et al., 1995; LaRue et al., 1995; Spackman and Hughes, 1995; Van der Haegen and Degraaf, 1996; Ewins, 1997; Kinley and Newhouse, 1997; Whitaker and Montevecchi, 1997; Meiklejohn and Hughes, 1999; Whitaker and Montevecchi, 1999; Whitaker et al., 2000), and mammals (Brusnyk and Gilbert, 1983; Servheen, 1983; Unsworth et al., 1989; Leach and Edge, 1994; Van der Haegen and Degraaf, 1996; Collins and Helm, 1997; Darveau et al., 1998; Darveau et al., 2001; Forsey and Baggs, 2001; but see De Groot, 2002). One underlying objective in riparian management has been to translate the spatial extent of riparian processes and patterns into management practice particularly buffer widths.

The use of riparian buffers has a long history in forestry. Implementation of treed corridors along waterbodies dates back to the 1700s in European forest management (Porter, 1887). The practice of leaving buffers was first applied in United States in late 1960s (Calhoun, 1988 reference in Brosofske et al., 1997). The primary reasons for the use of buffers today are similar to their historical use. For many jurisdictions, the underlying objective is the isolation of upland activities from terrestrial nearshore and aquatic areas. Despite the similarity of purpose, jurisdictions vary widely in the guidelines used in applying buffers. Variances in buffer widths could reflect differences in the integration of ecological, economic, and social factors. As an example, mountainous regions could be more likely to emphasize slope and drainage area in guidelines. Jurisdictions also face differing degrees and types of public and stakeholder scrutiny and economic incentives. Regions vary in the levels of competing interests in the forested lands such as aboriginal, recreation, rural home development, or fisheries. The complexity of guidelines could also reflect the interests of these groups as well as the response of mangers to demonstrate due diligence through greater guideline complexity. For most jurisdictions, the resultant riparian guidelines are a process of weighing all these factors and devising a compromise amongst often conflicting values for riparian areas.

Our primary objective is to review and analyze the structure and underlying riparian values embodied in forest management guidelines throughout jurisdictions in Canada and the United States. It is not an examination of the how effective these guidelines are in maintaining riparian values; this would require an examination of empirical data on water quality and aquatic and terrestrial habitat, and biota. Instead, this paper focuses on buffer width guidelines as one manifestation of resource management used to maintain riparian values. Specific objectives include: (1) comparison of national and regional differences in buffer widths, (2) comparison of modifying factors in structuring guidelines amongst regions, and (3) comparison of guidelines associated with harvest within buffers.

2. Data and analytical methods

A database of riparian management guidelines and regulations was obtained by contacting provinces, territories, and states in Canada and the United States (see Appendix A). We focused on jurisdictions which were able to provide a published record of riparian guidelines. Arizona, District of Columbia, Kansas, New Mexico, and Nunavit were not able to provide these and were not included in this paper. A total of 60 jurisdictions were analyzed. To examine the effect of broad regional geography on riparian guidelines, we categorized jurisdictions into six regions (Table 1). A number of different references were used to create the classification (Bailey and Cushwa, 1981; Environment, 2001; United States Department of Agriculture, 2002). Alaska, Alberta, British Columbia, and Washington were represented in two or more geographic regions. Washington state has separate guidelines for the eastern and western areas of the state. Statistical analysis required at least three jurisdictions in

Table 1
Classification of provinces, territories, and states from Canada and the United States into broad ecological regions

Country	Regions	Jurisdictions
Canada	Boreal	Alberta, Manitoba, Newfoundland, Northwest Territories, Ontario, Quebec, Saskatchewan, Yukon, British Columbia
	Northeast	New Brunswick, Nova Scotia, Prince Edward Island
	Rocky /Intermountain	Alberta, British Columbia
	Pacific	British Columbia
United States	Boreal	Michigan, Minnesota, Wisconsin, Alaska
	Rocky/Intermountain	Colorado, Montana, Utah, Wyoming, Idaho, Nevada, Washington east
	Midwest	Illinois, Indiana, Iowa, Missouri, Nebraska, North Dakota, Oklahoma, South Dakota, Texas
	Northeast	Connecticut, Delaware, Maine, Maryland, Massachusetts, New Hampshire, New Jersey, New York, Ohio, Pennsylvania, Rhode Island, Vermont, West Virginia
	Pacific	Alaska, California, Hawaii, Oregon, Washington west
	Southeast	Alabama, Arkansas, Florida, Georgia, Kentucky, Louisiana, Mississippi, North Carolina, South Carolina, Tennessee, Virginia

Table 2
Standardized template of waterbody types used to facilitate comparisons of guidelines amongst provinces, territories, and states

Waterbody types	Basic description	Size	Slope (%)	Fish-bearing	Drainage basin size
Large permanent stream	Permanent watercourse with defined bank, year-round flows	>5 m width	2.5	No	>50 km^2
Small permanent stream	Permanent watercourse with defined bank, year-round flows	≤ 5 m width	2.5	No	<50 km^2
Intermittent stream	Permanent watercourse with defined bank, no year-round flows	Any width	2.5	No	Not applicable
Small lake	Standing waterbodies	<4 ha	2.5	No	Not applicable
Large lake	Standing waterbodies	>4 ha	2.5	Yes	Not applicable

each category. Within the southwestern region (New Mexico, Arizona, and Nevada) only Nevada had published riparian guidelines, therefore this region was not included in analyses and Nevada was placed in with the Rocky/Intermountain region.

In order to compare waterbody classifications from different jurisdictions, we applied a standardized template of waterbody types (Table 2) to each jurisdiction. The criteria for the template were based on a preliminary review of all guidelines. Buffer widths and other management prescriptions from each jurisdiction were applied to each waterbody type. In the case of jurisdictions with relatively simple buffer guidelines, one or more classes often had the same buffer width. Other jurisdictions with more complex guidelines would often have to be re-interpreted. As an example, Wisconsin classifies streams as 'navigable', we interpreted this as large permanent streams. All subsequent analyses and descriptions of buffer widths were based on the template waterbody types. All buffer widths were reported in metric values.

Guidelines often included factors that modify the baseline buffer width assigned to a waterbody (e.g. presence of fish). The diversity and relative frequencies of different modifying factors were described for all jurisdictions. The use of complementary (i.e. two or more) modifying factors was examined by recording the paired frequency of factors and comparing this to the expected frequency based on independent selection. We also explored the changes to buffer widths associated with the application of five different modifying factors; waterbody type, waterbody size, slope, presence of fish, and selective harvest. Analysis of slope and presence of fish was limited to medium or large streams. Our purpose was to demonstrate general patterns of change to buffer widths rather than to exhaustively catalogue all combinations of factors to all types of waterbodies in all jurisdictions.

Tests of normality distributions on buffer widths indicated that datasets were slightly right skewed, however, all were within range of a normal distribution. As such, we applied a single factor analysis of variance (ANOVA). If significant differences were found, then multiple comparisons were detected using a Tukey Kramer HSD test. Of interest to managers is whether the combinations of factors are selected independently or whether modifying factors are selected by specific combinations. To test for biases in combinations of modifying factors, the probability of independent selection of factors was based on multiplying their individual occurrence within different jurisdictions to determine the random co-occurrence. This was compared to their actual co-occurrence amongst jurisdictions. A chi-squared test was used to compare expected (i.e. random) and actual occurrence. Throughout all statistical tests, a 5% probability was used as a criterion for significance. All analyses were executed on the JMP Statistical Program ver. 4.0.2 (SAS, 2000).

3. Results

3.1. Comparison of Canadian and American jurisdictions

Mean buffer widths varied from 15.1 to 29.0 m for each waterbody type for all jurisdictions from Canada and the United States combined (Table 3). In general, the pooled values from jurisdictions in the United States exhibited narrower buffer widths than in Canada for similarly classified waterbodies (Table 3). For most waterbodies, the mean buffer widths were 33–58% larger across

Table 3
The mean (S.E.) buffer widths summarized for jurisdictions from Canada and the United States combined, and separately for each country. Letters denote significant differences for waterbody types between Canada and the United States (ANOVA, d$f = 1$, post hoc Tukey Kramer HSD test, d$f = 1$, $P < 0.05$)

Waterbody classes	Combined ($n = 60$)	Canada ($n = 12$)	United States ($n = 48$)
Large permanent streams	28.1 (2.7)	43.8 (9.1) a	24.2 (2.3) b
Small permanent streams	21.8 (1.7)	29.6 (4.9) a	19.9 (1.7) b
Intermittent streams	15.1 (1.7)	13.8 (3.2) a	15.5 (2) a
Small lakes	27.6 (3.0)	47.1 (10.9) b	22.9 (2.1) b
Large lakes	29.0 (3.2)	54.6 (11.4) a	22.7 (2.1) b

Table 4
Comparison of regional differences among mean (S.E.) buffer widths for different waterbody types. Letters denote significant differences (ANOVA, df = 1, post hoc Tukey Kramer HSD test, df = 1, $P < 0.05$)

Waterbody types	Boreal ($n = 13$)	Rocky/Intermountain ($n = 9$)	Pacific ($n = 6$)	Northeast ($n = 16$)	Midwest ($n = 9$)	Southeast ($n = 11$)
Large permanent streams	39.1 (5.6) a	24.4 (7.2) ab	24.3 (8.0) ab	29.7 (7.2) ab	25.7 (5.9) ab	19.4 (3.0) b
Small permanent streams	26.3 (2.6) a	24.2 (7.2) ab	22.7 (7.9) ab	23.7 (4.1) ab	14.4 (1.2) b	17.5 (2.7) b
Intermittent streams	13.9 (3.0) ab	24.2 (7.2) a	21.7 (8.0) ab	13.1 (3.1) ab	11.5 (1.9) b	12.1 (3.4) ab
Small lakes	45.8 (8.2) a	23.0 (6.8) ab	22.7 (3.5) ab	30.6 (7.2) ab	21.7 (5.6) b	17.4 (2.8) b
Large lakes	52.2 (8.8) a	23.0 (6.8) ab	22.7 (3.5) b	30.2 (7.2) ab	21.7 (5.6) b	17.4 (2.8) b

Canadian jurisdictions. The exception was intermittent streams where there was no significant difference (Table 3).

3.2. Regional patterns

There were significant differences in the width of buffers across the different ecological regions. In general, the Boreal region had the widest buffers for all waterbody types except intermittent streams (Table 4). Mean Boreal buffer widths ranged from 13.9 m for intermittent streams to 52.2 m for large lakes. In contrast, the Southeast region had the narrowest mean widths ranging from 12.1 m for intermittent streams to 19.4 m for large streams. Rocky/Intermountain and Pacific regions had relatively little variance amongst waterbody types (±3 m). Both these regions had the widest buffers on intermittent streams. Northeast and Midwest also had relatively little variance amongst types except for intermittent streams which had buffers about half the width of other waterbody types.

3.3. Modifying factors

Across many jurisdictions, a number of modifying factors were commonly used in guideline formulation (Table 5). Just under half (44%) in North America had three or more modifying factors in the guidelines. Thirteen jurisdictions surveyed (22%) used only a single factor. Across all jurisdictions, waterbody type, slope, waterbody size, and presence of fish were the most common modifying factors (Table 5). Less common factors included: drinking water/aesthetics, drainage basin area, forest management practices adjacent to waterbodies, presence of saltwater flow, types of shoreline vegetation, upstream of fishbearing waterbodies, threat of downstream sediment transport, and flow rates.

Both Boreal and Pacific regions had the most diverse set of modifying factors (Table 5). Of the twelve most common factors, Pacific jurisdictions utilized a total of 11 with a mean of 4.8 factors per jurisdiction, while Boreal jurisdictions utilized nine with a mean of 3.5 factors per jurisdiction. Overall, Northeast and Rocky/Intermountain jurisdictions utilized a similar number of factors (8 and 9, respectively) as the Boreal, but jurisdictions within each of these regions featured means of 2.0 and 2.4 factors, respectively. Lastly, Midwest and Southeast guidelines had a relatively low number of factors, 5 and 6, respectively. Mean numbers of factors in these regions were 2.1 and 2.5 factors per jurisdiction, respectively.

Of the jurisdictions that used more than a single modifying factor, the most common combinations of two

Table 5
Mean number of modifying factors and the percentages of jurisdictions using different modifying factors assessed across all jurisdictions and each region

Modifying factor	All	Boreal ($n = 13$)	Rocky/Intermountain ($n = 9$)	Pacific ($n = 6$)	Northeast ($n = 16$)	Midwest ($n = 9$)	Southeast ($n = 11$)
Mean no. per jurisdiction	2.7	3.5	2.4	4.8	2.0	2.1	2.5
Waterbody type	78.7	91.7	77.8	66.7	68.8	88.9	100.0
Slope	49.2	25.0	44.4	50.0	43.8	66.7	63.6
Waterbody size	32.8	58.3	33.3	50.0	25.0	33.3	27.3
Fishbearing	32.8	58.3	33.3	83.3	18.8	11.1	36.4
Drinking water/aesthetics	14.8	16.7	22.2	33.3	18.8	11.1	9.1
Drainage basin area	6.6	25.0	0.0	0.0	6.3	0.0	0.0
Shoreline forest management	8.2	33.0	11.1	16.7	6.3	0.0	0.0
Saltwater flow	9.8	8.3	0.0	33.3	6.3	0.0	18.2
Shoreline vegetation	4.9	25.0	11.1	16.7	0.0	0.0	0.0
Upstream of fishbearing	4.9	0.0	9.1	16.7	6.3	0.0	0.0
Downstream sediment threat	3.3	0.0	0.0	33.3	0.0	0.0	0.0
Flow rates	3.3	0.0	0.0	33.3	0.0	0.0	0.0

Table 6
Most frequent combinations (pairs) of modifying factors used by jurisdictions. *N* represents the number of jurisdictions utilizing a particular combination. The expected percentage is based on independently selecting combinations of modifiers from the frequencies in Table 5

Combination of modifiers	*N*	Actual percentage	Expected percentage
Waterbody type—slope	23	39.0	38.7
Waterbody type—waterbody size	18	30.5	25.8
Waterbody type—fishbearing	18	30.5	25.8
Slope—fishbearing	11	18.6	16.1
Waterbody size—fishbearing	9	15.3	10.8
Fishbearing—drinking water/aesthetics	7	11.9	4.9
Waterbody type—drinking water/aesthetics	7	11.9	11.6

factors were waterbody type with either slope, waterbody size, or presence of fish (Table 6). Combinations of waterbody type with these modifiers were found in >30% of the guidelines examined. With the exception of waterbody type and slope, and waterbody type and drinking water/aesthetics, all other common combinations were utilized more frequently than would be expected based on independent selection of modifying factors (Table 6). That is, managers appear to be selecting combinations of modifiers to complement each other.

3.4. Waterbody types

Not surprisingly, streams were the most commonly recognized waterbody types across all guidelines (Table 7). Under half of jurisdictions (~39%) further recognized differences between intermittent and permanent flow streams. Lakes and wetlands were the third and fourth most recognized classifications, respectively. Less frequently used types included: waterbodies with exceptional aesthetics or heritage value, ponds, natural springs, saltwater/brackish estuaries, coldwater/warmwater bodies, braided streams, and manmade impoundments and canals.

Across all provinces, territories, and states, a mean number of 2.5 waterbody types were recognized per jurisdiction. Pacific, Boreal, and Southeast jurisdictions had the greatest diversity of waterbody types (8 or 9) and the recognition of 3.0–3.7 waterbody types per jurisdiction (Table 7). Rocky/Intermountain guidelines recognized a mean of 2.7 waterbody types per jurisdiction, with a total diversity of six waterbodies types. Both Northeast and Midwest jurisdictions recognized the least number of waterbody types, 1.8 and 1.9 per jurisdiction, respectively, and a total diversity of 5 and 4 waterbody types, respectively.

3.5. Slope

Results suggested that jurisdictions that do not incorporate shoreline slope as a modifying factor had wider baseline buffers to account for the potential presence of a sloped shoreline. On the other hand, jurisdictions that had specific guidelines for slope had narrower baseline buffers when there was no slope than jurisdictions without slope guidelines. The mean (S.E.) buffer width at 0% slope for jurisdictions with slope modifiers (16.8 m (2.9)) was significantly narrower than jurisdictions without slope guidelines (33.1 m (3.0); ANOVA, $P < 0.05$). Furthermore, in jurisdictions with slope guidelines, the mean (S.E.) addition to the baseline buffer width was 0.79 m (0.08) for each 1% increase in slope. Based on this relationship, we can crudely estimate the degree to which jurisdictions without slope guidelines extend their buffer width. A mean additional buffer of 16.3 m (33.1 − 16.8 m) could potentially account for 21% of slope change in jurisdictions without slope guidelines (Table 8). Northeast, Rocky/Intermountain, Pacific, and Boreal jurisdictions with no slope guidelines had the widest baseline buffers (Table 8). Across these regions, differences between baseline buffers of jurisdictions with and without slope guidelines varied from 18.6 to 29.7 m and accounted for 26–35% of slope. In

Table 7
Mean number of delineated waterbody types per jurisdiction and percentages of waterbody types assessed across all jurisdictions and by regions

Waterbody types	All	Boreal ($n = 13$)	Rocky/Intermountain ($n = 9$)	Pacific ($n = 6$)	Northeast ($n = 16$)	Midwest ($n = 9$)	Southeast ($n = 11$)
Mean no. per jurisdiction	2.5	3.0	2.7	3.7	1.8	1.9	3.2
Streams	69.7	84.6	77.8	83.3	43.8	66.7	90.9
Permanent/intermittent	39.4	38.5	33.3	50.0	18.8	22.2	90.9
Lakes	39.4	69.2	55.6	83.3	25.0	22.2	9.1
Marshes/bogs/wetlands	18.2	30.8	33.3	33.3	0.0	11.1	18.2
Aesthetics/heritage	10.6	15.4	11.1	16.7	12.5	0.0	9.1
Ponds	6.1	15.4	0.0	0.0	6.3	0.0	9.1
Estuaries	6.1	7.7	0.0	33.3	0.0	0.0	9.1
Natural springs	4.5	7.7	11.1	16.7	0.0	0.0	0.0
Cold/warmwater flows	3.0	7.7	0.0	0.0	0.0	0.0	9.1
Braided streams	1.5	0.0	0.0	0.0	0.0	0.0	9.1
Manmade waterbodies	1.5	0.0	0.0	16.7	0.0	0.0	0.0

Table 8
Buffer widths (m) from large streams when shoreline slope is used as a modifier for determining width. Baseline widths are the mean (S.E.) values at 0% slope for jurisdictions with slope guidelines. The No Guideline column is the mean (S.E.) buffer widths for jurisdictions without slope as a modifying factor. The rate column is the change in the mean additional buffer width with each percent increase in slope for jurisdictions with slope guidelines. Slope Accounted (%) represents the amount of slope that could be accounted by the wider baseline buffers in jurisdictions without slope guidelines

Region	Baseline width with guidelines (m)	Rate (m/%)	No. guideline width (m)	Difference (m)	Slope accounted (%)
All	16.8 (2.9)	0.79	33.1 (3.0)	16.3	21
Boreal	30.2 (4.4)	0.62	48.8 (6.9)	18.6	30
Rocky/Intermountain	17.7 (2.7)	0.73	43.2 (9.5)	25.5	35
Pacific	18.0 (3.6)	1.12	47.7 (11.8)	29.7	26
Northeast	15.4 (1.3)	0.79	39.6 (9.0)	24.3	31
Midwest	14.0 (2.1)	0.77	16.5 (2.3)	2.5	3
Southeast	10.1(1.0)	0.56	19.8 (4.2)	9.6	17

contrast, Southeast and Midwest regions exhibited a much lower difference between jurisdictions with and without slope guidelines. Buffers from Southeast and Midwest jurisdictions without slope could potentially account for slopes of 17 and 3%, respectively.

3.6. Waterbody size

Table 9 summarizes the mean ranges used to delineate size classes within waterbody types. Relatively few jurisdictions further classified waterbody types into different size categories beyond defining minimum size criteria. Only 30, 10, and 10% of jurisdictions established further size classes for streams, lakes, and wetlands, respectively. In pooling all jurisdictions, we found three size categories of streams, two size categories of lakes, and two size categories of wetlands. In applying mean values to the breakpoints of these size classes, we estimated that small streams were <5.0 m in width, medium streams were between 5.0–9.3 m while large streams were >9.3 m. Small lakes were >0.9–4.3 ha, while large lakes were >4.3 ha. Small wetlands were 0.3–2.3 ha, while large wetlands were >2.3 ha. Boreal, Rocky/Intermountain, and Pacific region jurisdictions featured the greatest number of divisions by size class within waterbody types. Amongst this group there were relatively few differences between size class boundaries except the Boreal region, which classified large wetlands as >5.0 ha. Both the Midwest and Southeast regions featured three classes for streams, a single class for lakes, and no delineation for wetlands (Table 9). Lastly, the pooled data for the Northeast region indicated two stream classes, and a single class each for lakes and wetlands (Table 9).

It is worth noting that most jurisdictions have some categories reserved for areas of exceptional value, usually historic or natural sites. These areas usually have much wider buffers. Furthermore, a number of jurisdictions ($n = 7$) do not use direct measurements of channel width or surface area as criteria for separating waterbody types. In these jurisdictions, size was part of the formulation but other criteria were also considered. West Virginia and Saskatchewan base their classifications on stream order rather than a direct metric of channel width, while Wisconsin utilizes channel navigability. The Northwest Territories emphasizes the terrestrial and riparian interface as well as floodplain width to classify streams. A number of maritime jurisdictions, (i.e. New Brunswick, Newfoundland, and Nova Scotia), base their classifications on mapping units such as delineation on 1:50,000 maps.

3.7. Presence of fish

Like those for slope guidelines, jurisdictions with fish guidelines have narrower baseline buffers than those without fish guidelines (Fig. 1). Across all jurisdictions, the mean (S.E.) baseline buffer widths for large non-fishbearing streams in jurisdictions that utilize fishbearing modifiers was 18.5 m (4.9), however, there was a significant increase to 45.7 m (6.4) for large streams with fish (ANOVA, $P < 0.05$). In comparison, jurisdictions without

Table 9
Mean waterbody size classes for streams (m), lakes (ha), and wetlands (ha) summarized across all jurisdictions and for each region

Waterbody type	All	Boreal	Rocky/Intermountain	Pacific	Northeast	Midwest	Southeast
Small streams	0.1–4.9	0.4–3.0	0.2–3.7	0.3–3.8	0.0–5.3	0.0–6.1	0.0–6.3
Medium streams	5.0–9.3	3.1–4.3	3.8–5.0	3.9–5.0	>5.3	6.2–12.2	6.4–12.2
Large streams	>9.3	>4.3	>5.0	>5.0		>12.2	>12.2
Small lakes	0.9–4.3	2.1–.5	2.8–5.0	0.4–5.0	>1.1	>0.0	>0.0
Large lakes	>4.3	>7.5	>5.0	>5.0			
Small wetlands	0.3–2.3	0.3–5.0	0.4–3.5	0.6–3.5	>0.0		
Large wetlands	>2.3	>5.0	>3.5	>3.5			

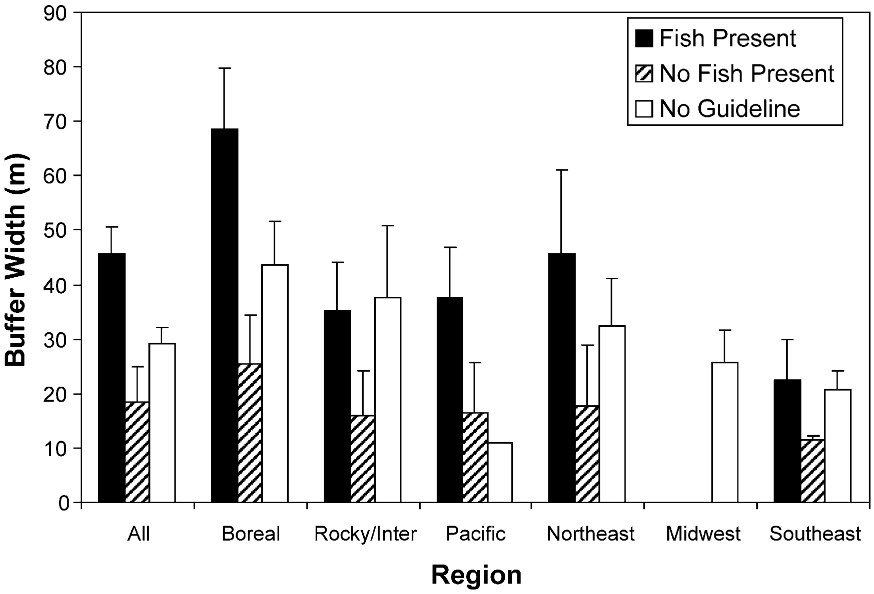

Fig. 1. Mean buffer widths of large streams with fish (first bar) and without fish (second bar) for jurisdictions with fish guidelines, and jurisdictions without fish guidelines (third bar). Error bars represent standard error.

fish guidelines had an intermediate mean baseline (S.E.) buffer of 29.1 m (3.1). There appears to be some compensation for not having fish guidelines by maintaining wider baseline buffers in jurisdictions without fish guidelines. This pattern was present in all regions except the Midwest, where there were no jurisdictions with fish guidelines and for the Pacific where all jurisdictions except Hawaii had fish guidelines (Fig. 1).

3.8. Patterns of selective harvest

About 80% of all jurisdictions allowed some harvest within buffers. Unlike slope or fish guidelines, most jurisdictions added no additional buffer width to areas that permitted harvest within buffers. The mean (S.E.) width of buffers amongst jurisdictions permitting harvesting was 27.4 m (2.9). Surprisingly, jurisdictions that did not allow harvesting had slightly wider buffers, 34.3 m (5.5). Amongst regions, all jurisdictions in the Midwest allowed for selective harvest within buffers, while 62% of Boreal jurisdictions allowed harvest. The remainder of regions were ordered Pacific (83%), Northeast (75%), Southeast (73%), and Rocky Mountain/Intermountain (67%). Like the pooled dataset, there were no clear patterns amongst regions in terms of buffer widths for jurisdictions with and without selective harvest. Jurisdictions with selective harvest in Boreal, Northeast, and Pacific regions had mean buffers wider than jurisdictions without selective harvest, while in Rocky/Intermountain and Southeast jurisdictions the reverse was true (Fig. 2). In neither case were any of the differences statistically significant (ANOVA, $P > 0.05$).

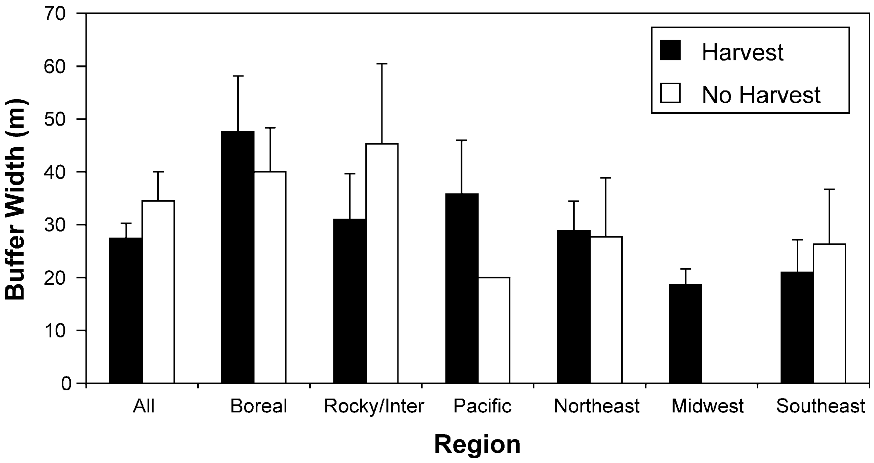

Fig. 2. Mean buffer widths on large streams for jurisdictions with selective harvest (first bar) and jurisdictions without selective harvest (second bar). Error bars represent standard error.

4. Discussion

From an ecological perspective, the primary goal for managers is to match or exceed the width of guidelines to those recommended in the ecological literature. Fortunately, a large body of research literature on riparian function, structure, and biota is available with many studies putting forward recommendations on buffer width. Both theoretical and empirical data are available for sedimentation and erosion control, and stream temperature regulation. As an example, Wong and McCuen (1982) modeled the impacts of substrate characteristics, slope, vegetation roughness, and overland flow patterns and suggested that under most circumstances, buffers less than 60 m were sufficient to control sedimentation. Applying a different model, Cook College Department of Environmental Resources (1989) found that buffer widths from 15 m for slopes less than 1% and 61 m for slopes under 15% would be adequate for sediment reduction. Results from field studies suggest more varied results. Moring (1982) reported that 30 m buffers were unable to prevent increases in stream sedimentation after a partial clearcutting of small watersheds in Oregon. In contrast, Haupt and Kidd (1965) found that 9 m strips were sufficient to remove sediments from cutblock features in Rocky Mountain areas of central Idaho. Low relief boreal systems are unlikely to yield large amounts of sediment even without buffer strips. Steedman and France (2000) found no significant sediment deposition after shoreline harvesting to small coldwater lakes in the Canadian Shield.

In a review of forest practices, Binkley and Brown (1993) noted that almost universal use of intact or partially harvested buffers had significantly reduced increases in stream temperature after harvesting. A number of models exist for prescribing buffer widths to control solar radiation (Beschta and Weatherred, 1984). The important variables include: stream width and volume, buffer forest height and density, amount of watershed cut, solar inputs, and groundwater temperatures. These models suggest that buffers >30 m wide are sufficient to prevent steam temperatures from rising. Aubertin and Patric (1974) found that partially cut ($\sim 50\%$ retention) buffers (10–20 m wide) were still able to sufficiently shade streams and prevent temperature increases. Removal of buffers from patch cut and clearcut/burned basins in the H.J. Andrews Experimental Forest in the western Cascades, Oregon, increased the maximum summer temperature earlier in the season (Johnson and Jones, 2000).

Other empirical studies indicate that the removal of buffers from stream systems causes increases in water temperature and light, and subsequent changes in aquatic biota. In a comparison of undisturbed and partially harvested (26–54% tree removal, 6–17 years previously) stream segments in northeast Oregon, Carlson et al. (1990) reported macroinvertebrate densities 20–113 times greater in logged sites, although diversity was the same between logged and undisturbed sites. The increased densities were particularly notable in lower elevation streams and those less shaded by vegetation. Similarly, greater amounts of light after logging led to increases in the density of both invertebrate and periphyton communities in small, buffered (8–9 m) streams in northern New England (Noel et al., 1986). Newbold et al. (1980) studied the impact of varying stream buffer widths on invertebrate communities in northern California streams. Streams with buffers ≥ 30 m exhibited no impact of harvest on invertebrates, while streams with buffers <30 m experienced changes in species diversity. For the narrower buffers, the changes in diversity were positively correlated to buffer width. Young et al. (1999) found that non-anadromous cutthroat trout streams harvested to streamside margins reached a maximum summer temperature of 30 °C, which was correlated with a four-fold decline in fish density. Fish populations recovered after stream temperatures decreased following revegetation by shrubs and trees. Wesche et al. (1987) found that overhead bank cover provided by riparian vegetation explained the greatest amount of variation in trout population size in Wyoming streams.

In stream and possibly small lake systems, most of fine and large organic debris is derived from treed riparian buffers. France et al. (1996) found that harvesting of the riparian canopy around Boreal Shield lakes reduced the allochthonous inputs of small woody debris by 90%. Similar declines in allochthonous materials have been noted in a number of forested, small stream systems in which treed riparian buffers have been removed (Bilby and Bisson, 1992; Duncan and Brusven, 1985). Treed riparian areas are the source for large woody debris in stream ecosystems (reviewed in Sedell et al., 1990). A number of studies have demonstrated that most ($>90\%$) of the coarse woody inputs originated within a single tree length of the stream bank (Robinson and Beschta, 1990; Van Sickle and Gregory, 1990). In Washington and Oregon, a 30 m strip on one side of a stream provided 85% of the natural uncut input, but a 10 m strip provided less than half the natural input (McDade et al., 1990).

In general, buffer widths recommended for protection of terrestrial riparian components were wider than those recommended for aquatic components. Harper and MacDonald (2001) demonstrated that a distinct lakeshore forest edge community extended for about 40 m around boreal lakes in central Alberta. Johnson and Brown (1990) compared the forest composition of buffers strips (~ 80 m wide) left after timber harvest to the composition of undisturbed lakeshore forests in Maine. They reported that shrub densities were greater in the buffer strips, but tree and snag densities were greater in undisturbed lakeshore forests. A comparison of different widths and disturbance levels of riparian buffers in mature balsam fir forests in Quebec indicated that narrower buffers (20 m intact and 20 thinned) exhibited greater stem densities of conifer and deciduous shrubs than wider buffers (>40 m) and uncut controls (Darveau et al., 1998).

With respect to maintenance of riparian bird assemblages, Spackman and Hughes (1995) argued that buffer widths of 150 and 175 m would be required to maintain 90 and 95% of preharvest species along mid-order streams in Vermont. Kinley and Newhouse (1997) argued for riparian buffers wider than 50 m in order to maintain bird densities and prevent changes to community structure in hybrid white × Engelmann spruce-lodgepole pine forests of southeastern British Columbia. A comparison of buffer strips (~80 m) left after timber harvest and undisturbed lakeshore areas in Maine indicated that density and species richness were lower in the buffer strip (Johnson and Brown, 1990). In contrast, Whitaker and Montevecchi (1999) found that buffer strips contained a higher total avian abundance than forested streamside controls. This was attributed to the presence of edge and clearcut tolerant species. Riparian buffers were able to maintain many riparian and woodland species, however, interior forest species required wider buffers and some were not present in even the widest buffers (~60 m). Darveau et al. (1995) noted an increased bird density in streamside buffers the year after cutting balsam fir forests of Quebec. These differences declined in following years. Density increases were greatest and subsequent declines were faster in narrower (20 m) buffers. However, thinning (33% tree removal) of 20 m strips did not appear to have as much of an impact on bird species as reducing buffer width did. They suggested that buffer widths of 60 m could support forest dwelling species, whereas buffer widths of 20 m were more useful to ubiquitous species.

For most large mammals, buffers left after harvest are not wide enough to provide source habitat, however, they may provide sufficient cover for foraging and travel. In west-central Idaho, Unsworth et al. (1989) recommended that forested buffers along streams, roads, and dense stands on north-facing slopes be retained for bear cover and bedding. Van der Haegen and Degraaf (1996) found that black bears used riparian buffers as travel corridors in harvested stands in Maine. Brusnyk and Gilbert (1983) found that moose densities were greater in riparian buffer strips (60 m) left after harvesting than in blocks that did not retain buffers.

In contrast, the impact of riparian buffers on small mammals appears variable. Forsey and Baggs (2001) found that track counts were greater for Newfoundland marten, snowshoe hare, and red squirrel in interior, uncut forests than riparian areas, whereas track counts were greater in riparian strips (20 m) after cutting. The authors concluded that buffer strips left after harvest were valuable to these species. Darveau et al. (2001) studied small mammals in balsam fir forests along streams in Quebec and found no difference in the abundance of the two most common small mammal species among buffers of varying width (20, 40 and 60 m). They also reported that meadow vole, which was absent prior to harvest, invaded clearcuts and was a limiting factor to the occurrence of red back vole and deer mouse in buffer strips. They suggested that 20 m buffer strips may work as refuges for small mammals, but that wider strips would provide a more natural habitat for edge-avoiding species. Contrary to these results, a number of studies found little bias in the distribution of mammals in buffers left after harvest. In comparing red squirrel, northern flying squirrel, and eastern chipmunk population parameters within upland and riparian strips and forested blocks, Cote and Ferron (2001) found no differences among treatments and controls. De Groot (2002) found similar results for small mammals within mixedwood boreal forests in north-central Alberta. Abundances and demographics of red-backed voles, deer mice, and meadow voles estimated through trapping did not differ in riparian forest strips (20–200 m) and controls adjacent to small lakes up to four years after their creation. In balsam fir forests of Quebec, Darveau et al. (1998) reported that snowshoe hares made only minimal use of riparian buffer strips regardless of width (widths of 20, 40 and 60 m were tested).

While it is beyond the scope of this paper to thoroughly review all literature (e.g. Wenger, 1999), the wider recommendations associated with some terrestrial species may reflect a direct loss of habitat with reductions in buffer width. In contrast, habitat loss may only occur with very narrow buffers for some aquatic organisms, e.g. amphibians. Viewed in this light, jurisdictions with wider buffers would tend to capture a greater extent of terrestrial riparian functions, structure, and biota. Twenty-nine (48%) jurisdictions make explicit statements about the protection of aquatic and terrestrial habitats and biota in either their preamble or objectives for riparian management. The remaining jurisdictions focus on the protection of aquatic habitats and biota. Based on the ecological literature, the former group should have wider buffers. However, buffer widths were not statistically different between the two groups for any of the waterbody types (ANOVA; P range 0.22–0.90). This suggests that application of wider buffer widths does not necessarily follow from a desire to expand protection to the terrestrial components of the riparian.

One of the more striking patterns in this study was the variance amongst jurisdictions in the complexity of guidelines. In general, jurisdictions seemed to select between management paradigms that either apply relatively simple, guidelines with few factors or more complex guidelines that utilize a large number of factors. Jurisdictions such as those in the Pacific region have been more proactive in the development of complex guidelines. In contrast, most of the jurisdictions in the Midwest retained relatively simple guidelines. It would be tempting to argue that the greater intricacy in guidelines reflects the greater complexity in the ecological setting of Pacific jurisdictions. In part this maybe true, however, many of modifying factors, waterbody types, and size categories found within Pacific jurisdictions are general enough that they could be applied to other jurisdictions. Viewed in this light, the added complexity may not necessarily stem from an inherently more complex underlying riparian ecology.

One possible explanation may be that jurisdictions in the Pacific region, particularly those of continental North America, support significant forest-based economies both in terms of timber harvest and other non-extractive uses such as recreation. All these jurisdictions have been the focus of intense public and regulatory scrutiny over the past few decades. In general, the political response of government and industry has been to produce more complex guidelines. Modifying factors can be interpreted as a priority list for riparian protection. Slope, presence of fish, drinking water, and aesthetics were relatively frequent across all regions. For some jurisdictions in the Boreal, drainage basin area, shoreline vegetation, and types of forestry activity were also utilized. Pacific regions included these and upstream and downstream effects on fish and sediment and saltwater flows. In designing complex guidelines, a common pattern was to use the major classifiers (waterbody size and type) to develop baseline widths and then increase buffer widths when special factors were present. Our analysis suggested that jurisdictions increase guideline complexity by adding specific combinations of modifiers usually by the addition of slope, fishbearing, or drinking water to other more common modifiers (Table 6). The question of whether this approach has resulted in improvements to the management of riparian habitat and biota remains a point of contention and requires empirical data based on large-scale experimentation.

In the case of slope and fishbearing streams, buffer widths used by jurisdictions without these modifying factors were intermediate to those with modifying factors. In essence, these jurisdictions treated all waterbodies as potentially being bordered by slope or bearing fish. In the case of slope, application of a safety margin through additional width maybe warranted. Relationships between slope, buffer width, and sediment transport are monotonic and relatively continuous (Wong and McCuen, 1982). Additional buffer width produces a relatively predictable result in terms of a safety margin for sediment transport. In particular, waterbodies may be better protected by wider buffers from periodic disturbances such as unusually wet years, catastrophic weather, or catastrophic disturbances to upland areas. During these years there is the potential for greater amounts of run-off and erosion potential. Wider buffers may reduce the risk of sediment transport into waterbodies. A number of stream classification systems utilize the occurrence of high flows as a basis of categorization (e.g. Rosgen, 1996).

The relationship between buffer width and presence of fish is less straightforward. Changes in buffer width may cross multiple thresholds such as those for the input of coarse woody debris (McDade et al., 1990; Robinson and Beschta, 1990; Van Sickle and Gregory, 1990) or regulation of stream temperature (Brazier and Brown, 1973). It is unclear whether those jurisdictions without fish guidelines had established relationships between local fish populations/communities and buffer widths. In this case, the application of a wider baseline buffer may not provide an incremental increase in protection unless the additional width crosses a threshold value.

The finding that Canada has wider buffers than the United States was somewhat surprising. In part, it can be explained by the high proportion of Boreal jurisdictions in Canada and Southeast jurisdictions in the United States. Boreal jurisdictions had some of the widest buffers while Southeastern jurisdictions had some of the narrowest. A number of possibilities exist for differences amongst these regions. Generally, more populated regions have a longer history of development in riparian areas. If much of the riparian has historically been allocated to development, it is much more difficult to apply larger buffers that may take away from an already established user group. Also, areas with longer histories of development may have already significantly altered riparian habitats. Hence, the range of riparian values may be significantly changed or lost.

Most jurisdictions ($\sim 80\%$) allowed for the option of harvest within riparian buffers. Types of harvest included single tree selection, group selection, and zoned harvest. Though jurisdictions differed in prescriptions, the general restrictions were similar throughout. These included: (1) retaining at least half the cover, volume, or basal area, (2) minimizing or eliminating machinery traffic, and other ground disturbance, (3) protecting understory and regeneration, (4) preventing direct shoreline erosion or removal of trees with roots that stabilized shorelines, (5) spatially dispersed cutting (single tree or small group selection), and (6) preventing 'hi-grading' of large or exceptional timber value trees.

Harvest within buffers attempts to extract some direct, short-term economic benefit from riparian areas and re-introduce or maintain tree-replacing disturbances. Although this can be viewed as contrary to the longstanding riparian management paradigm of protection through preservation, partial harvest has been argued as a management analogue for natural single tree or small group replacement. Ilhardt et al. (2000) and Palik et al. (2000) argue that partial harvest would fit into a probabilistic model for defining transverse riparian values. They suggest that riparian structure, function, and biota are more likely to be found closer to the water's edge. Palik et al. (2000) further suggests a gradient of decreasing harvest intensity with distance to water's edge follows this definition. Their model features a continuous gradient from no harvest areas to single tree selection, small group selection, and large group selection with retention as one moves from water's edge to the upland. From an ecological impact standpoint, partial harvest within buffers, if carefully executed, seems to have relatively little effect on potential short-term impacts such as stream temperature, however, long-term effects such as the potential reduction of large organic inputs have not been evaluated. In practice, a number of jurisdictions have multiple-management zones along some waterbodies (e.g. Washington, British

Columbia). These feature zones of differential harvest rather than a continuous change but still retain the same underlying principles. A no harvest zone closest to water edge, then a series of zones with increasing degrees of harvest as one moves to the upland.

4.1. Management implications

There are a number of trends in buffer guidelines that have management implications. The first is a shift away from 'one or few-sizes-fits-all' buffers. In part, this has been driven by a combination of economic incentives, improvements in best management practices for timber extraction primarily in skidding and road construction, increase in knowledge base, increase in public scrutiny, and a desire to protect the unique ecology of riparian areas. Resource managers potentially find themselves in an ever-shortening cycle of revising and implementing riparian management guidelines. Our results suggest that for most factors such as fishbearing and slope, one or few-sizes rules may potentially apply buffers that are wider than would be warranted by local site conditions. Hence, some would argue this applies inconsistent criteria to the delineation and protection of riparian values in the field.

The current trend has been towards more 'tailor-made' buffers that vary amongst broadly similar harvest areas to within a single harvest area. The primary benefit in using tailor-made buffers is the application of clear criteria to define the riparian. These criteria are specifically defined by the modifying factors selected by jurisdictions, and buffers are applied in a predictable response to these criteria. The caveat to tailor-made buffers is the greater complexity in guidelines. From the dataset of guidelines in this study, we found a total of 14 broad modifying factors and individual jurisdictions applied from 1 to 6 of these factors. For each factor, there can exist two to many classes. Hence, the addition of a single factor can exponentially increase the number of potential classes applied in the field. The upper limit to the number of classes is often set by the practicality of training personnel, costs with planning and prescribing modifiers in the field, and the compliance requirements and monitoring of buffers. In particular, compliance monitoring is generally considered much easier with simple guidelines. In discussions with many resource managers from both regulatory and industry sides, these issues greatly favor the application of simpler guidelines.

A second trend in the application of riparian buffers is their use within a broader watershed framework. The naturally integrative nature of watersheds and their natural segregation within the landscape makes them attractive tools for land management. The application of buffers classified at a stand level and repeated throughout a watershed may not meet broader objectives. A number of issues require a broader perspective for riparian management. These include: aquatic and terrestrial travel corridors for biodiversity, habitat fragmentation, cumulative effects, and downstream water quality and human consumption. At their earliest inception, buffers were largely used to protect aquatic resources. Increasingly, they are viewed as an important component for the maintenance and dispersal of upland species and other ecological values such as old growth or dispersal corridors. Aquatic components have always had a strong research foundation using the watershed framework (e.g. Naiman et al., 1987). We would argue that the current overall emphasis on stand-level prescriptions is due to the underlying principle of aquatic protection for buffers. If aquatic components are isolated from upland activities through buffers then there is no need to adjust guidelines for watershed effects at least as modifications to buffer guidelines. Other impacts such as water yield and peak flow impact are driven by the percentage of watershed harvested rather than buffer area (Keenan and Kimmins, 1993). Extension of riparian management to terrestrial components requires integrating riparian and upland ecological processes and biota. Research on the watershed implications for terrestrial components has accumulated over the last decade (e.g. Knopf and Samson, 1994; Naiman and Rogers, 1997; Lock and Naiman, 1998) but has not yet been translated into comprehensive guidelines that integrate buffer widths at the watershed scale.

5. Conclusions

The overall goals of riparian protection through the use of buffers meets the ecological recommendations for most aquatic and some terrestrial components of the riparian. Most notably core habitat for medium and large mammals and birds were wider than most current guidelines. In these cases, more research would be required to determine how changes in buffer width alter the overall habitat quality for these biota. It could be argued that the variance amongst jurisdictions in the width of buffers suggests emphasis on differing riparian components. In part, these reflect broad differences in ecoregions. However, other correlates such as the history of land use, degree of public scrutiny, and framework for the guidelines themselves contributes to the overall variance. With the last point, jurisdictions choose between having simple 'one or few sizes fits all' or relatively complex guidelines that considers modifying factors such as the presence of fish, slope, and other factors. The number of potential classes for riparian buffers greatly increases with the addition of even a few modifying factors. In the future, two management trends are likely to continue, the shift towards more complicated guidelines and the expansion to larger-scale, watershed planning of riparian areas.

Table A1
Reference list of guidelines from different jurisdictions in Canada and the United States used in this paper

Jurisdiction	Reference
Alabama	Alabama Forestry Commission (Undated). *Alabama's Best Management Practices for Forestry*. Montgomery, Alabama. 28 p. Alabama Forestry Commission. http://www.forestry.state.al.us/publication/BMPs/BMPs.pdf (accessed 2002)
Alaska	Division of Forestry, Department of Natural Resources. (2000). *Alaska Forest Resources and Practices*. Anchorage, Alaska. 22 p. Division of Forestry, Department of Natural Resources. http:/www.dnr.state.ak.us/forestry/pdfs/forprac.pdf (accessed 2002)
Arkansas	Arkansas Forestry Commission. (Undated). *2.0 Streamside Management Zones*. Little Rock, Arkansas. 3 p. Arkansas Forestry Commission. http:/www.forestry.state.ar.us/bmp/smz.html (accessed 2002)
California	California Department of Forestry and Fire Protection. (2000). *California Forest Practice Rules 2000*. Sacramento, California. 230 p. Resource Management, Forest Practice Program, California Department of Forestry and Fire Protection. http://fire.ca.gov/forest_practice.html (accessed 2002)
Colorado	Colorado State Forest Service. (1998). *Colorado Forest Stewardship Guidelines to Protect Water Quality: Best Management Practices (BMPs) for Colorado*. Fort Collins, Colorado. 32 p. Colorado State Forest Service, Colorado State University
Connecticut	Connecticut Resource Conservation and Development Forestry Committee. (1998). *A Practical Guide for Protecting Water Quality While Harvesting Forest Products*. Hartford, Connecticut. 36 p. Connecticut Resource Conservation and Development Forestry Committee, Department of Environmental Protection, State of Connecticut
Delaware	Delaware Department of Agriculture, Forest Service. (1996). *Delaware's Forestry Best Management Practices Field Manual*. Dover, Delaware. 71 p. Delaware Department of Agriculture, Forestry Department
Florida	School of Forest Resources and Conservation. (Undated). *Special Management Zones*. Gainesville, Florida. 13 p. Florida Forestry Information, School of Forest Resources and Conservation, University of Florida. http://www.sfrc.ufl.edu/Extension/ffws/smz.htm (accessed 2002)
Georgia	Georgia Forestry Commission. (1999). *Georgia's Best Management Practices for Forestry*. Dry Branch, Georgia. 71 p. Georgia Forestry Commission
Hawaii	Division Forestry and Wildlife. (2001). *Water Protection and Management Program*. Honolulu, Hawaii. 23 p. Division Forestry and Wildlife, Department of Land and Natural Resources, State of Hawaii
Idaho	Idaho Department of Lands. (1996). *State of Forestry for Idaho-Best Management Practices: Forest Stewardship Guidelines for Water Quality*. Coeur d'Alene, Idaho. 33 p. Idaho Department of Lands, Bureau of Forestry Assistance
Illinois	Illinois Department of Natural Resources. (2000). *Forestry Best Management Practices for Illinois*. Springfield, Illinois. 63 p. Division of Forest Resources, Department of Natural Resources
Indiana	Indiana Department of Natural Resources. (1999). *Indiana Logging and Forestry Best Management Practices, BMP Field Guide*. Indianapolis, Indiana. 85 p. Department of Natural Resources, Division of Forestry
Iowa	Iowa Department of Natural Resources. (1998). *Iowa Forestry: Best Management Practices*. Des Moines, Iowa. 65 p. Iowa Department of Resources. http://www.state.ia.us/government/dnr/organiza/forest/bmps3.htm (accessed 2002)
Kentucky	Division of Forestry. (1997). *Kentucky Best Zones. Management Practice No. 3-Streamside Management*. Frankfort, Kentucky. 47-55 pp. Division of Forestry, Department of Natural Resources. http://www.ca.uky.edu/agc/pubs/for/for67/bmp_03.pdf (accessed 2002)
Louisiana	Louisiana Department of Agriculture and Forestry. (1999). *Recommended Forestry Best Management Practices for Louisiana*. Baton Rouge, Louisiana. 83 p. Louisiana Department of Agriculture and Forestry. http://www.ldaf.state.la.us/divisions/forestry/publications.asp (accessed 2002)
Maine	Maine Department of Environment Protection. (1998). *A Field Guide to Laws Pertaining to Timber Harvesting in Organized Areas of Maine*. Augusta, Maine. Publication DEPL W39-B98. 35 p. Maine Forest Service, Department of Conservation Maine Forest Service. (1994). *Erosion and Sediment Control Handbook for Main Timber Harvesting Operations. Best Management Practices*. Augusta, Maine. Publication SHS#22. 48 p. Forest Information Centre, Maine Forest Service, Maine Department of Conservation Maine Department of Environment Protection. (1999). *Maine Shoreland Zoning: A Handbook for Shoreland Owners*. Augusta, Maine. Publication DEPL W1999-2., 34 p. Maine Department of Environmental Protection
Maryland	Maryland Department of Natural Resources—Forest Service. (2000). *A Guide To Maryland Regulation of Forestry and Related Practices*. Annapolis, Maryland. 81 p. Maryland Department of Natural Resources. http://www.dnrweb.dnr.state.md.us/download/forests/frg.pdf (accessed 2002)
Massachusetts	Kittredge, Jr., D.B. and Parker, M. (1996). *Massachusetts Forestry Best Practices Manual*. Pittsfield, Massachusetts.56 p. Bureau of Forestry, Division of Forests and Parks, Department of Environmental Management, Commonwealth of Massachusetts
Michigan	Michigan Department of Natural Resources. (1994). *Water Quality Management Practices on Forest Land*. Lansing Michigan. 9 p. Forest Management Division, Michigan Department of Natural Resources
Minnesota	Minnesota Forest Resources Council. (1999). *Sustaining Minnesota Forest Resources-Voluntary Site-level Management Guidelines for Landowners, Loggers, and Resource Managers. Part 3. Integrated Guidelines*. St Paul, Minnesota. 78 p. Minnesota Forest Resources Council
Mississippi	Mississippi Forestry Commission. (2000). *Mississippi's BMP's: Best Management Practices for Forestry in Mississippi*. Jackson, Mississippi. 15 p. Publication # 107 (Internet Version). Mississippi Forestry Commission, http://www.mfc.state.ms.us/pdf/bmp2000.pdf">http://http://www.mfc.state.ms.us/pdf/bmp2000.pdf (accessed 2002)
Missouri	Missouri Department of Conservation. (1997). *Missouri Watershed Protection Practice*. Jefferson City, Missouri. 29 p. Missouri Department of Conservation

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Jurisdiction	Reference
Montana	Department of Natural Resources and Conservation. (1993). *Montana Guide to the Streamside Management Zone Law and Rules*. Missoula, Montana. 35 p. Department of Natural Resources and Conservation, Department of Natural Resources and Conservation
Nebraska	Nebraska Forest Service. (Undated). *Forestry: Best Management Practices for Nebraska*. Lincoln, Nebraska. 6 p. School of Natural Resource Sciences, University of Nebraska. http://www.ianr.unl.edu/pubs/forestry/nfs/nfs-1.htm (accessed 2002)
Nevada	State of Nevada. (1997). *Nevada Forest Practice Regulations (Statutes) for Forestry. Chapter 528 Forest Practice and Reforestation NRS 528.053. Certain activities prohibited near bodies of water; Nevada Revised Statutes*. Carson City. Nevada. 528-8 pp. Nevada State Legislature. http://www.leg.state.nv.us/lawl.cfm (accessed 2002)
New Hampshire	New Hampshire Division of Forests and Lands. (Undated). *Forest Operations Manual*. Concord, New Hampshire. 31 p. New Hampshire Division of Forests and Lands. http://www.nhdfl.com/for_mgt_bureau/manual/Forest%20Operations%20Manual.pdf (accessed 2002)
New Jersey	New Jersey Forest Service. (Undated). *New Jersey Forestry and Wetlands Best Management Practices Manual*. Jackson, New Jersey. 31 p. Forest Resource Education Center
New York	Division of Lands and Forests. (Undated). *Timber Harvesting Guidelines*. Albany, New York. 4 p. Division of Lands and Forests, New York State Department of Environmental Conservation. http://www.dec.state.ny.us/website/dif/privland/privassist/bmp.html (accessed 2002)
North Carolina	North Carolina Division of Forest Resources. (1989). *Forestry Best Management Practices Manual*. Raleigh, North Carolina. 67 p. North Carolina Division of Forest Resources, Department of Environment, Health, and Natural Resources Department of Environment, Health, and Natural Resources. (1990). *Best Management Practices for Forestry in the Wetlands of North Carolina*. Raleigh, North Carolina. 28 p. Department of Environment, Health, and Natural Resources
North Dakota	North Dakota State Forest Service. (1999). *North Dakota Forestry Best Management Practices*. Bottineau, North Dakota. 29 p. North Dakota State Forest Service
Ohio	Ohio Division of Forestry. (Undated). *Fact Sheet: Best Management Practices for Logging Operations*. Columbus, Ohio. 4 p. Division of Forestry Publications, Ohio Division of Forestry. http://www.hcs.ohio-state.edu/ODNR/Education/pdf/logging.pdf (accessed 2002)
Oklahoma	Oklahoma Cooperative Extension Service. (1998). *Riparian Area Management Handbook*. Stillwater, Oklahoma. Publication E-952. 96 p. Oklahoma Cooperative Extension Service, Oklahoma State University
Oregon	Oregon Department of Forestry. (2002). *Division 635 Water Protection Rules: Purpose, Goals, Classification and Riparian Management Areas. Oregon Administrative Rules 629-635-0000 to 629-635-0310*. Salem, Oregon. 10 p. Oregon State Archives, Oregon Secretary of State. http://www.arcweb.sos.state.or.us/rules/Rules/fpa-635.htm (accessed 2002)
Pennsylvania	Division of Forest Advisory Services. (1999). *Inventory Manual of Procedure for the Fourth State Forest Management Plan*. Harrisburg, Pennsylvania. 49 p. Bureau of Forestry, Department of Conservation and Natural Resources, Commonwealth of Pennsylvania
Rhode Island	Rhode Island Forest Conservators Organization. (Undated). *Best Management Practices for Rhode Island. Water Quality Protection and Forest Management Guidelines*. North Scituate, Rhode Island. Rhode Island Forest Conservators Organization. (accessed 2002)
South Carolina	South Carolina Forestry Commission. (1994). *Best Management Practices: Streamside Management Zones*. Columbia, South Carolina. 4 p. South Carolina Forestry Commission. http://www.state.sc.us/forest/rbsmz.htm (accessed 2002) South Carolina Forestry Commission. (Undated). *Best Management Practices for Braided Systems: A Supplement to the 1994 BMP Manual*. Columbia, South Carolina. 5 p. South Carolina Forestry Commission. http://www.state.sc.us/forest/braid.htm (accessed 2002)
South Dakota	South Dakota Department of Agriculture. (Undated). *South Dakota Forestry Best Management Practices-Forest Stewardship Guidelines for Water Quality*. Rapid City, South Dakota. 32 pp. Resource Conservation and Forestry, South Dakota Department of Agriculture
Tennessee	Division of Forestry. (1993). *Guide to Forestry Best Management Practices*. Nashville, Tennessee. 41 p. Division of Forestry, Tennessee Department of Agriculture
Texas	Texas Forest Service. (2000). *Texas Forestry Best Management Practices*. College Station, Texas. 108 p. Texas Forest Service
Utah	State of Utah, Non-Point Source Task Force. (1998). *Nonpoint Source Management Plan-Silvicultural Activities*. Salt Lake City, Utah. 92 p. Division of Forestry, Fire, and State Lands, Department of Natural Resources
Vermont	Vermont Agency of Natural Resources. (1987). *Acceptable Management Practices for Maintaining Water Quality on Logging Jobs in Vermont*. Waterbury, Vermont. 51 p. Vermont Agency of Natural Resources, Department of Forests, Parks and Recreation
Virginia	Virginia Department of Forestry. (Undated). *Forestry BMP guide for Virginia*. Charlottesville, Virginia. 31 p. Virginia Department of Forestry. http://state.vipnet.org/dof/wq/bmpguide.htm (accessed 2002)
Washington	Washington Forest Practices Board. (2000). *Washington Forest Practices Board Manual: Section 7 Guidelines for Riparian Management Zones*. Olympia, Washington. 44 p. Washington State Department of Natural Resources. http://www.wa.gov/dnr/htdocs/fp/fpb/fpbmanual/se07.html (accessed 2002)
West Virginia	Center for Agricultural and Natural Resources Development. (Undated). *Best Management Practices-Soil and Water Conservation*. Morgantown, West Virginia. 3 p. Center for Agricultural and Natural Resources Development. West Virginia University Extension Service. http://www.wvu.edu/~agexten/forestry/bestprac.htm (accessed 2002)
Wisconsin	Wisconsin Department of Natural Resources. (1997). *Wisconsin's Forestry: Best Management Practices for Water Quality-Field Manual*. Madison, Wisconsin. 76 p. Bureau of Forestry, Wisconsin Department of Natural Resources. http://www.dnr.state.wi.us/org/land/forestry (accessed 2002)

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Jurisdiction	Reference
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Appendix A

See Table A1.

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